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Examining Avian Diversity in Acadia National Park Through Time

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EXAMINING AVIAN DIVERSITY IN ACADIA NATIONAL PARK THROUGH TIME

by

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ABSTRACT

Ecosystems experience change due to both natural causes and anthropogenic impact such as habitat fragmentation and climate change. Avian species are used as habitat indicators to observe ecosystem integrity and have been observed to experience changes in biodiversity due to anthropogenic impact. This study examines the temporal and spatial changes of avian biodiversity in Acadia National Park. We seek to understand (1) how the alpha diversity has changed over time on Mount Desert Island and Schoodic Peninsula, (2) how beta diversity has changed over time for Mount Desert Island and Schoodic Peninsula (3) how the Schoodic Woods Campground can be used as a model for avian biodiversity change due to human impact. This study demonstrates that the avian communities of Acadia National Park have experienced change. We found that for both Mount Desert Island and Schoodic Peninsula the alpha diversity and beta diversity have increased over time. Comparing Mount Desert Island to Schoodic Peninsula over time resulted in a decrease of beta diversity. Although alpha diversity exhibited significant change surrounding the Schoodic Woods Campground, an increase in species richness closer to surface edges for trails, and campground, there was no trend for beta diversity. The observed trends could be due to biotic homogenization as well as edge effect leading to increased levels of biodiversity.

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INTRODUCTION

Ecosystems, defined as communities with both living and non-living components, change throughout time (Gurevitch et al, 2006; Tansley, 1934). Some change occurs naturally, such as earthquakes, precipitation, and succession. However, human impact, also known as anthropogenic change, has readily increased the rate and intensity at which ecosystem change is occurring worldwide (Gaffney and Steffen, 2017). There is a wide variety of ways that humans influence ecosystem health, including land use change, ecological effects anthropogenic climate change, overexploitation of natural resources, introduction of invasive species, and pollution for example (Matthews, 2014).

1.1 Climate Change

One of the major ways humans are exacerbating ecosystem changes is through anthropogenic climate change. Climate change is a global or regional long-term change in climate. Climate change can be exacerbated by natural events such as changes in solar radiation, tectonic activity, volcanic eruptions and El Niño effects (Ricke and Caldeira, 2014). However, recently the rate of climate change has accelerated because of anthropogenic effects on the environment (Gaffney and Steffen, 2017). The increase of industrial technology and dependence on fossil fuels have increased the concentration of greenhouse gases in earth's atmosphere, which has led to a number of changes scientists have observed in earth's climate. Evidence suggests that globally, earth's average temperature is warming, and sea level is rising (NOAA, 2018; Settele *et al*, 2014; Vose *et al*, 2004). Regionally, effects of climate change vary, including changes in precipitation regime, increased wild fires, increased intensity and frequency of extreme weather events, and decreased ice cover (Matthews, 2014). Climate change is a global occurrence.

Climate change has a significant impact on all types of biota in the world. It has been demonstrated to impact species ranges, abundance, diversity, phenology and behavior (Hughes, 2000; Settele *et al,* 2014). Generally, the habitat range are shift towards the poles and higher in elevation as increasingly warmer temperatures lead to unsuitable habitat conditions (Chen *et al,* 2011; Hughes, 2000; Parmesan and Yohe, 2003). For example, a study done on North American bird species, found that northern limit of birds with southern distributions shifted northward by 2.35 km a year (Hitch and Leberg, 2007). Climate change has variable effects on species abundance and causes some species to increase and some to decrease in population number (Parmesan and Yohe, 2003). A study done on birds demonstrated that generalists are likely to increase, and specialists are likely to decrease in abundance due to climate change (Davey *et al,* 2012). Generalist species are those that are able to live in a wide variety of habitats, while specialized species are those that are only able to live in a specific area. Climate change can lead to increased levels of biotic homogenization, functional similarity of two of more biotas over a specific time interval (Olden, 2006; Savage and Vellend, 2014). The climatic effect on biodiversity is also variable, especially on a local level. However, global trends exhibit an overall decrease in biodiversity (Bellard *et al¸* 2012). Scientists have observed that the Earth is most likely entering into a sixth mass extinction demonstrated by predicted 21% to 52% species as taxa becoming extinct (Barnosky *et al,* 2011). Climate impacts the behavior and phenology in diverse ways (Parmesan and

Yohe, 2003). Although greatly variable, climate change has significant effects on biota on multiple scales.

1.2 Human Development

Anthropogenic impact on ecosystems can be both indirect, i.e. climate change, and direct, i.e. human development. Human development can be in the form of residential and commercial expansions, logging, developing infrastructure and agriculture. Human development affects components of ecosystem health (Matthews, 2014). Human development can lead to effects similar to those observed by climate change but on a smaller scale. Human development can lead to increased habitat fragmentation, the division of larger habitats into smaller, more isolated habitats, and destruction in addition to increased edge effects, changing populations due to abrupt edges (Didham, 2010). Habitat destruction is connected to decreases in species abundances and diversities (Lehman *et al*, 1994). Habitat fragmentation, different from simple habitat destruction also affects biodiversity and abundance. Studies have shown that habitat fragmentation is generally linked to decreases in biodiversity and abundance (Fahrig, 2003; Wilson *et al,* 2016).

1.3 National Parks

One method that conservationists and environmentalists have employed to reduce the impact of humans on ecosystems, specifically by maintaining biodiversity, is through preserving land using the National Park Service. The first national park, Yellowstone National Park, was created in 1872 as a "public park or pleasuring-ground for the benefit and enjoyment of the people" while maintaining the ecological integrity and preserve of a natural land (National Park Service Overview, 2017). The entire National Park Service was created to “preserve unimpaired the natural and cultural resources and value of the National Park Service for the enjoyment, education and inspiration of this and future generations” (National Park Service Overview, 2017). Scientists have argued that the primary consideration for selecting potential national park land was beauty and uniqueness of the areas (Shafer, 1999). In 1906, the American Antiquities Act was created which allowed the United States President to set aside land with great scientific or historic interest as national monuments (Dorr, 1942). This act has helped maintain the ecosystem within the parks. A study done by Kathryn Miller demonstrated that there were significant levels of increased biodiversity and older successional forests in the eastern United States national parks compared to outside of them (Miller *et al,* 2016). In 1916, President Woodrow Wilson signed an act that established the National Park Service, which led to the further expanse of parks across the country (History U.S. National Park Service). Today, the National Park System owns and manages over 84 million acres of land with 417 sites (National Park Service Overview, 2017). The number of visitors has continually increased from 1 million visitors in 1920 to 331 million visitors in the centennial year 2016 (National Park Service Overview, 2017). The National Park Service has been important in maintaining culture, history and biological integrity of these natural areas while being available for recreation use.

1.4 Acadia National Park

Acadia National Park, primarily located on Mount Desert Island off the coast of Maine, was the first national park established east of the Mississippi River and has a strong history of preservation and admiration (Dorr, 1942). Prior to 1604, when Samuel Champlain ventured to Mount Desert Island from France, the Wabanaki people inhabited the island (History of Acadia, 2017). The first recorded natural history observations of what would become Acadia National Park occurred in 1880 when a group of students from Harvard including Charles Eliot and Edward Lothrup Rand came to the island for the summers to study plants, birds, insects, fish, geology, hydrology, and meteorology (Schmitt, 2014). Their early dedication eventually led to further preservation of the area. After President Theodore Roosevelt signed the American Antiquities Act into law in 1906, more people became interested in preserving the land. In 1916, the land that would later become Acadia National Park, was given to the federal government for protection by George Dorr and became known as Sieur de Monts National Monument (History of Acadia, 2017). Around this time, Henry Labe Eno noted that the Sieur de Monts National Monument was critical for avian preservation due to its geographical position along a major migratory path, its coastal features and physical character (Eno, 1916). In 1919, John D. Rockefeller Jr. donated his first of many parcels of land to the park and the monument became the first national park east of the Mississippi River, Lafayette National Park, to better protect the area for biological conservation and preservation of the land (Acadia National Park History, 2017). The land on Schoodic Peninsula was donated to the park with the requirement that it be the same year that the park was officially named Acadia National Park, 1929 (Workman, 2014). Additional land continued to be added to the park and additional preservation movements occurred until Acadia National Park became what it is today.

Acadia National Park continues to be of importance for ecological preservation and scientific research. The only national park in the Northeast, it provides a unique protected island habitat for the species that live there (Vauz *et al,* 2008). The park consists of over 35,000 acres owned by the National Park Service and over 12,000 acres of privately owned lands with conservation easements managed by the National Park Service (Park Statistics, 2015). Although most of the area of the park is located on Mount Desert Island, it is also partially located on Schoodic Peninsula, Isle au Haut and multiple smaller islands in the area. The habitat of Acadia National Park is unique. It is located in a biological transition zone between southern deciduous and northern deciduous forests with many sub habitats including wetlands, sub-alpine, meadows, salt marshes, and forests (National Park Service, 1992). The total area of Acadia National Park only makes up one percent of the total area in Maine, but it includes many rare plant species found in very few places (Greene, 2005). There is a great amount of biodiversity with over 150 locally rare plant species, at least 45 species of terrestrial mammals, 12 species of marine mammals. 17 species of amphibians, 5 species of reptiles and 338 species of birds (National Park Service, 1992). The unique habitat makes it an ideal place to study and understand habitat preservation.

The preservation of Acadia National Park cannot protect it from the stresses of anthropogenic impacts. According to Maine's Climate Future report, published the University of Maine, the average annual temperature of Maine has increased by 3˚F between 1895 and 2014 amidst year-to-year fluctuations (Fernandez *et al,* 2015). In addition, precipitation levels in Maine have increased by 6 inches since 1895 and snowfall has declined by roughly 15 percent (Fernandez *et al,* 2015). Similar warming trends have occurred within Acadia National Park. Within Acadia National Park, the eastern half of Mount Desert Island has the greatest increase in average temperatures, followed by Schoodic Peninsula (Gonzalez, 2014). Due to the large amount of summer visitors, Acadia National Park can have one of the highest concentrations of ozone in the entire state (Vaux *et al,* 2008). Anthropogenic impact has been shown to affect Acadia National Park in the past.

Acadia National Park has also observed changes in the overall patterns of biota that correspond with the climate change. Several studies demonstrate that Acadia's vegetation has always been changing (Fisichelli *et al*, 2013, 2015; Gonzalez, 2014; Harris *et al* 2012). In a study investigating the impact of climate change on forests, Fisichelli noted that in addition to an overall northern shift of tree species, there were 13 tree species with decreasing habitat and 18 species with increasing habitat (Fisichelli *et al,* 2013). Throughout the various national parks across the country there has been an overall change in the vegetation ranging between 22-77% tree species affected (Fisichelli *et al,* 2013). However, in Acadia National Park it was noted that over 70% of tree species were in large change categories directly related to climate change (Fisichelli *et al,* 2013). A study done by Caitlin McDonough MacKenzie found 15.8% of the flora recorded in the late 19th century by naturalists is now locally extirpated (McDonough MacKenzie, 2017). The increased levels of ozone in Acadia were shown to cause leaf damage and reduce the growth rate of plants (Vaux *et al, 2*008). Climate change has directly contributed to changes in plant abundance and distribution in Acadia National Park.

In addition to changes in plant species, climate change has also contributed to changes in the avian compositions (Bank *et al,* 2006; Gonzalez*,* 2014; Vaux *et al, 2008*). A recent study demonstrated that while the national parks are becoming increasingly important for avian communities, they are still experience change due to climate change, especially the northeast parks (Wu *et al,* 2018). Birds in Acadia National Park show various patterns in population changes.

Population Trends [D]			
Trend	*# Species*		
Increasing	18 (MDI); 12 (SCH)	4 (MDI); 1 (SCH)	8 (MDI); 6 (SCH)
Decreasing	3 (MDI); 2 (SCH)	0 (MDI); 1 (SCH)	2 (MDI); 6 (SCH)
No Change	23 (MDI); 23 (SCH)	0 (MDI); 2 (SCH)	13 (MDI); 14 (SCH)
Insufficient data	93 (MDI); 99 (SCH)	25 (MDI); 25 (SCH)	16 (MDI); 13 (SCH)

Figure 1 Population trends of land (column 2), marsh (column 3) and marine (column 4) bird species on Mount Desert Island and Schoodic Peninsula from Christmas Bird Counts (Vaux et al, 2008)

For example, 13% of species are increasing in abundance, and 2% are decreasing and there is deficiency in information for 68% of the species (Vaux *et al,* 2008). A study done on migrating breeding birds in Maine found that over 60 species had at least one significant effect of climate change and that there is a moderate relationship between changes in the avian community and climate change (Wilson, 2007).

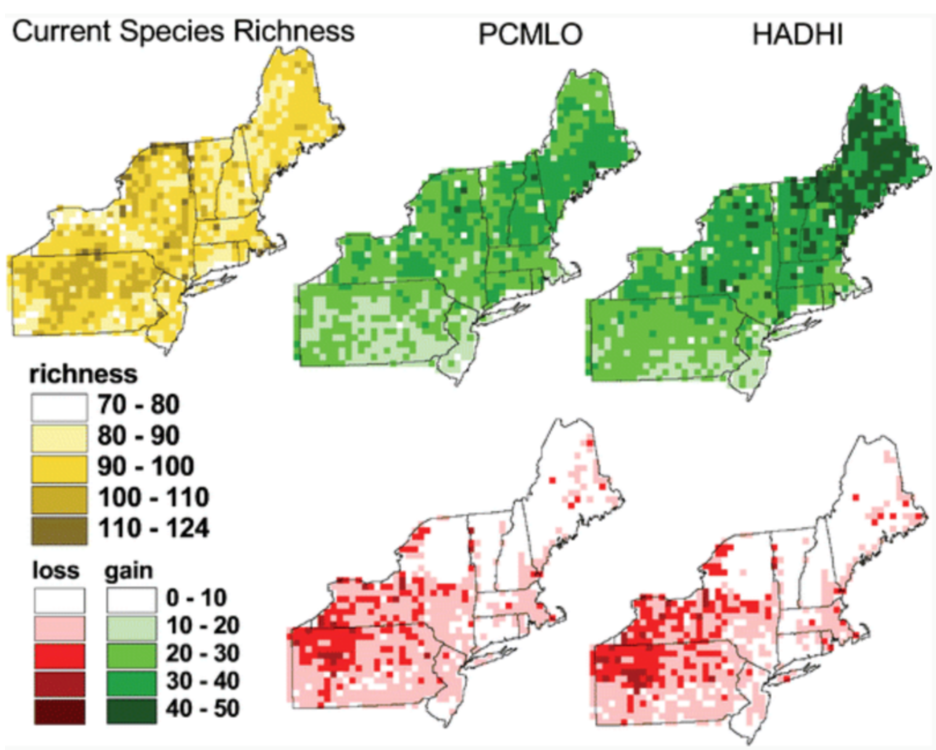

Figure 2 Projected gains and losses of bird species richness across the northeastern United States published in the study: Potential Effects of Climate Change on Birds of the Northeast (Rodenhouse et al, 2007)

In 2014, researchers with the Northeast Temperate Network found 77 avian species inhabiting the park, a lower count than previous years (Faccio and Mitchell, 2015). Another study found that under two climate change scenarios, least and most change, there was decreasing habitat suitability for 30 avian species, no change for 15 species and new habitat for 48 species (Fisichelli *et al,* 2014). In addition to abundance changes, Acadia National Park has also noted changes in habitat ranges. According to a study done examining the vulnerabilities of Acadia National Park due to climate change, winter bird ranges have shifted northward roughly 0.5 km a year from 1974 to 2004 (Gonzalez, 2014). Visitors have directly affected bird in Acadia National Park, especially loons, through nest disturbance and other factors (Vaux *et al,* 2008).

In addition to the changing climate, Acadia National Park has also experienced impacts due to human disturbance. Many people visit Acadia National Park every year, and the number is only increasing. In 2017, the park was one of the top ten visited national parks in the country with over 3,509,271 recreational visitors, an increase from the previous year (Annual Visitation Highlights, 2018). Recreational visits to Acadia National Park have generally been increasing in number since the park was established, as noted in the figure below (National Park Service Visitation Statistics, 2017). Similar trends have been observed on Schoodic Peninsula. According to reports published by the National Park Service between June and October, there were 167,211 recreational visits in 2009, 191,003 recreational visits in 2015 and 246,135 recreational visits in 2016 (Jacobi and Flesh, 2017; Jacobi and National Park Service, 2011). The significant increase in recreational visits from 2015 to 2016 is likely due to the development of new Schoodic Woods Campground mentioned above. The increasing number of visitors, leading to more required infrastructure and development, is likely going to have significant impacts on the overall ecosystem health and species diversity.

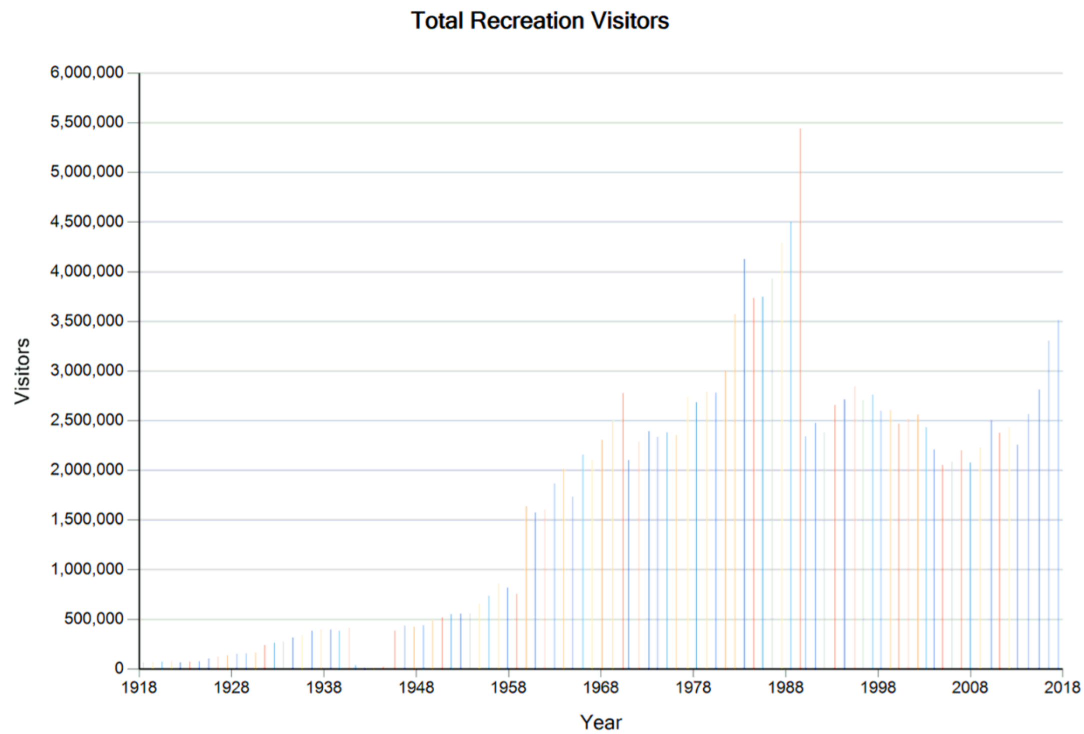

Figure 3 Increasing visitation trends in Acadia National Park (Visitor Use Statistics, 2018)

Visitors have directly affected the vegetation of Acadia National Park as vegetation cover on Cadillac Mountain decreased between 1979 and 2001 directly due to off trail trampling (Kim and Daigle, 2010). Additional changes to the vegetation in Acadia National Park have been observed due to invasive plants such as purple loosestrife, pests and pathogens, nitrogen deposition, mercury pollution and the fire of 1947 (Harris *et al,* 2012).

Figure 4 Change in land cover on the eastern part of Mount Desert Island (Google Earth, 2017)

Figure 5 Change in land cover of Schoodic Peninsula in 1985, 1997, and 2015 due to logging and a campground development (Google Earth, 2017)

Human development has also changed Acadia National Park, recently on Schoodic Peninsula. Although there are some signs of fishing, farming, and logging, Schoodic Peninsula has largely been undisturbed (Workman, 2014). Visitors to Schoodic Peninsula historically have represented only about 10% of all the visitors to Acadia National Park (Jacobi and National Park Service, 2011). However, recently there has been a change in the land use on Schoodic Peninsula as the Schoodic Woods Campground was developed in 2015 as noted above in figure 3 (Kelly and National Park Service, 2015). Schoodic Woods Campground is located three miles southeast of Winter Harbor. Around the campground, there is a six-mile loop road, over eight miles of hiking trails and eight miles of bike paths. The campground has 94 RV and tent sites available

(Kelly and National Park Service, 2015). Both Schoodic Peninsula and Mount Desert Island have experienced change due to anthropogenic impact.

1.5 The Importance of Birds

The impact of climate change and human development have regularly been studied using avian populations (Davey *et al,* 2012; Hitch and Leberg, 2006; Miller *et al,* 2010; Wilson, 2007). Studies have demonstrated that birds can be used as local species indicators for environmental health (Canterbury *et al,* 2000; Hatfield *et al,* 2017). The study done by Canterbury found that avian species was positively correlated with vegetation assemblage species richness (Canterbury *et al,* 2000). Climate change affects the distribution, abundance, diversity and phenology of bird populations. The most commonly observed impact of climate change for avian populations is a shift in migration patterns and timing (Millet *et al*, 2010). A study done by Herbert Wilson found that 101 avian species in North American demonstrated a difference in arrival patterns due to climate change, with the majority arriving earlier (Wilson, 2007). Avian populations are also exhibiting poleward habitat shifts as observed in the United States where the northern habitat limit of birds has moved northwards roughly 2.35 km a year (Hitch and Leberg, 2006). Similar trends were observed in the United Kingdom (Hitch and Leberg, 2006). Climate change affects avian abundance and diversity. A study done on Mount Kilimanjaro found that avian functional identity was directly dependent on the vegetation biodiversity and indirectly dependent on climate change (Vollstädt *et al,* 2017). The 2014 State of the Birds Report noted that 432 avian species, roughly one third of the overall species in the United States, are at the highest risk of extinction (North American Bird Conservation Initiative, 2014). Although overall avian biodiversity and abundance have decreased due to climate change, there are various local and regional affects. One study done in the United Kingdom has demonstrated that biodiversity might increase on a local level due to the increase of generalist species in a particular area (Davey *et al*, 2012). Climate change affects birds in a variety of different ways and therefore can be used as indicators of climatic effects.

In addition to climate change, human development has also influenced avian communities and species. A study done on bird communities in France using Breeding Bird surveys found that biotic homogenization is largely correlated to landscape disturbance and habitat fragmentation (Barnagaud *et al*, 2011; Devictor *et al* 2008). A study done examining how spatial distribution of birds have changed due to timber harvesting found that there was an abrupt change in bird species, followed by a slower population rebound. (Campbell *et al,* 2012). Agriculture is one significant way that humans have changed ecosystems. A study done in 2003 found that roughly a quarter of pre-agriculture birds were lost post agriculture (Gaston *et al,* 2003). Climate and human development both have significant impact on avian species.

This study aims to demonstrate how the biodiversity in Acadia National Park has changed over time due to anthropogenic impacts. To test for changes in the biodiversity of Acadia National Park, I focused on the avian community. Acadia National Park has hosted long-term monitoring efforts for birds, but changes in biodiversity have not been examined in this community. Although birds have been shown to be sensitive to various anthropogenic changes such as climate change and changes in land use, this study examines the temporal changes in the avian diversity in correlation to the observed climate change and anthropogenic impact, not causation. I tested the general patterns of change, but the specific drivers are unknown. However, I used spatial diversity in relation to the development of Schoodic Woods Campground as a case study to directly test the drivers, anthropogenic development, of the observed changes. The questions I investigated in this research include (1) how has the alpha diversity has changed over time on Mount Desert Island and Schoodic Peninsula both individually and in relation to each other, (2) how has beta diversity has changed over time for Mount Desert Island and Schoodic Peninsula both individually and in relation to each other and (3) how can Schoodic Woods Campground can be used as a model for avian biodiversity change due to human impact by measuring alpha and beta diversity. I predicted that Schoodic Peninsula and Mount Desert Island were increasing in similarity over time and that the development of the campground would lead to a decrease in the overall biodiversity of the area.

METHODS

Study System

The majority of Acadia National Park is located on Mount Desert Island, the largest island off the coast of Maine with 108 square miles of area (Britannica, 2017). However, four miles to the east as the crow flies is Schoodic Peninsula, the only section on the park found on the mainland. It is home to a loop road and the Schoodic Institute at Acadia National Park, a non-profit that conducts environmental research, education, and outreach. The United States Navy originally built the campus to use as a base, and then once returned to the National Park Service, it became Acadia Partners for Science and Learning in 2004 and Schoodic Institute in 2013 (Workman, 2014). Throughout the park's history, Schoodic Peninsula has been less developed and less visited compared to the portions of Acadia National Park that are on Mount Desert Island. Although once home to a U.S. Navy base, a few small towns and logging and fishing practices, Schoodic Peninsula is more forested than most areas of Mount Desert Island. Most of the land contained in Acadia National Park has been protected since it became a national park, in 1919 (Workman, 2014). The areas surrounding the park, while not directly protected by the National Park Service, have experienced minimized anthropogenic impact. The greatest anthropogenic impact likely comes from the development of new roads, housing areas and campgrounds, which may lead to increased foot and car traffic in addition to habitat disturbances. This study investigates how anthropogenic impact has influenced Acadia National Park.

Data collection

Temporal changes in biodiversity

I used the Audubon Society's Christmas Bird Count (CBC) surveys done by Audubon Society to analyze temporal change in biodiversity in Acadia National Park. Christmas Bird Count data is provided by the National Audubon Society and through the generous efforts of Bird Studies Canada and countless volunteers across the western hemisphere. The CBC is a national effort that began in 1900. The count for Mount Desert Island (MEMD) started in 1933 and for Schoodic Peninsula (MESP) started in 1956 (National Audubon Society, 2010). Christmas Bird Counts are conducted every year during a 24-hour period within two weeks of December 25th (National Audubon Society, 2010). It is important to note that only birds present in the winter, both resident and northern migrant, are observed in the Christmas Bird Count data and this can alter the overall understanding of the species distributions and abundances. This study uses all types of birds that were observed in Acadia National Park, rather than examining either land birds or sea birds. In 1950, the current protocols for data collection were established, although specific changes were not noted other than switching from group counts to individual counts (National Audubon Society, 2010).

Christmas Bird Counts are conducted using a variety of methods. Avian presence and abundance observations are made using both sights and sounds. Bird counts can be conducted by hiking, biking, boating, canoeing, snowmobiling, driving or sitting by a specific bird feeder and counting. Although there is an attempt to maintain the same methods of collection from year to year, there is no guarantee as routes and counters vary. Although the variation in method does have the potential to cause problems with data analysis, there have been numerous studies done that use Christmas Bird Counts as the primary source of data. For example, there have been studies done investigating large scale diversity (Bock C.E and Bock J.H, 1974; Murthy, 2016; Rybicki, 2007), distribution (Bock and Lepthien, 1976; Garrison, 1993; James and Ethier, 1989) and abundance (Kendeigh, 1937; Lepthien and Bock, 1976; Pandolfino 2012). It is important to consider the variation in methods and that the data isn't a reliable substitution for detailed scientific studies, but provides information on larger more general trends. According to a review examining the scientific value of the Christmas Bird Count, using presence-absence data and using a correction or bias factors are way to mitigate the problems caused by the variation in the methodology (Dunn *et al,* 2005). This particular study used absence-presence measures instead of abundance for this reason. However, no correction factor was implemented. The variation in methodology presents a challenge but can be minimized when using Christmas Bird Count data.

In Acadia National Park, CBCs are conducted annually in two areas, Mount Desert Island and Schoodic Peninsula. Each CBC covers a 15-mile diameter circle with minimized overlap. One of the major problems noted with using Christmas Bird Count data in scientific research is that the count circles aren't random and are usually centered around population areas (Dunn *et al,* 2005). However, this isn't a consideration for this particular study because only two specific count circles are investigated. The center of the circle for Mount Desert Island is half a mile east of Halls Quarry at Somes Sound and the center point for Schoodic Peninsula is the junction of Route 186 and Summer Harbor

Road in South Gouldsboro (National Audubon Society, 2010). A map is included below for references of the specific areas covered.

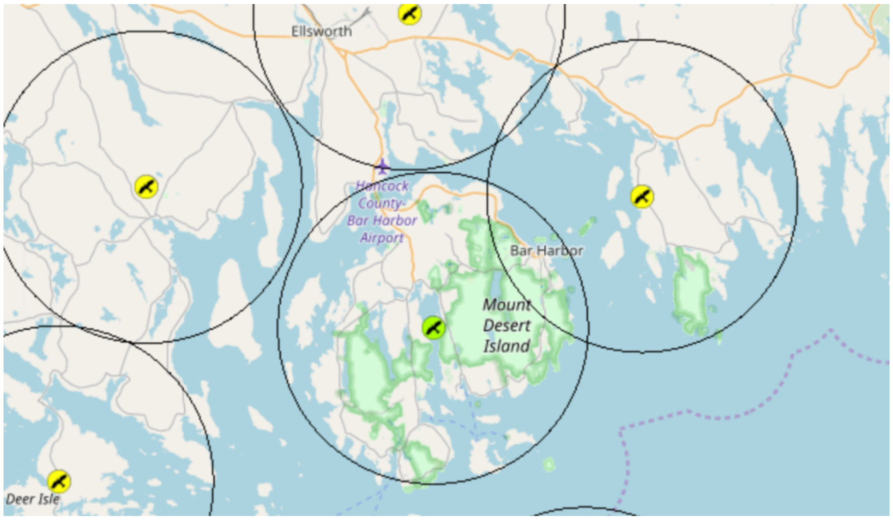

Figure 6 Christmas Bird Count perimeter circles, specifically Mount Desert Island (lower center) and Schoodic Peninsula (right) and center points (National Audubon Society)

According to the Schoodic Peninsula Christmas Bird Count complier, Seth Benz, over the past few years for the Christmas Bird Counts on Schoodic Peninsula, the routes have been the following (personal communication):

Route Number	Description
1	Hancock Point from Cross Road to the Point (both east and west sides of point itself)
2	Waukeg Neck (all of Sorrento)
3	Route 1 including Punkinville, Thorne, Tunk Lake, Ashbille Spur, and Guzzle Roads
4	Route 186 from Route 1 to Winter Harbor including Grindstone Neck and side roads
5	Winter Harbor Town
6	Schoodic Point and Schoodic Environmental Research Center (SERC) Campus
7	Little Moose Island to Bunkers Harbor to Birch Harbor to Route 186 back to Winter Harbor
8	Birch Harbor to Prospect Harbor to Corea
9	Paul Bunyan Shores, Gouldsboro Bay
10	Pond Road, West Bay Road (east half of Route 186)

Table 1 Schoodic Peninsula Christmas Bird Count routes

There is not a set list of routes that are counted on Mount Desert Island. However, according to Michael Good of Downeast Nature Tours, the complier of the Mount Desert Island Christmas Bird Count, most of the coastal habitat and some of the interior habitat was generally covered (personal communication). Areas typically covered include Hulls Cove, Sand Point Emmery Cottages, Salsbury Cove, Mount Desert Biological Laboratory, Hadley Point, the Oceanarium and Town Hill. The specific methodology is also variable from year to year and across both Schoodic Peninsula and Mount Desert Island. According to Seth Benz, the compiler of the Schoodic Peninsula Christmas Bird Count, in 2017, 4 to 6 different groups consisting of 1 to 4 people covered the circle (personal communication). Each group was assigned a route in the morning and another route in the afternoon. The methodology used for data collection on Mount Desert Island is not as detailed for Schoodic Peninsula, but likely very similar with many counts being completed by boat or car.

<u>Spatial changes in biodiversity</u>

I used previously gathered avian abundance data from Schoodic Ecosystem Systems Project (SchESP) acquired by Katharine Ruskin and University of Maine undergraduate students to test the spatial changes in biodiversity as a direct result of anthropogenic impact. The Schoodic Ecosystem Services Project data was collected following the protocols used in the Northeast Temperate Monitoring Network inventory (Faccio *et al,* 2015). Ten-minute passive point counts were conducted at 46 locations along transects throughout Schoodic Peninsula. Transects were designed to measure edge effect, so they started at edges of human development, such as the Schoodic Woods Campground, and moved into the surrounding forest. The counters visited each point four times throughout the breeding season (late May to mid-July). There were point count stations along transects around the campground spaced approximately 250 meters apart. The counts lasted ten minutes and the approximate distances, 0-10m, 10-25m, 25-50m and >50m, were recorded. The researchers observed the abundance and presence of birds using both sight and sound.

Figure 7 Google Earth image of the Schoodic Woods Campground and surrounding hiking and biking trails in 2011 (left) and 2015 (right) demonstrating the new development (Google Earth, 2017)

Changes in Avian Biodiversity

Species biodiversity is often used as indicator for ecosystem health and a measure of change. In this research, I used species diversity to understand how Schoodic Peninsula and Acadia National Park have changed comparatively over time. There are three main types of biodiversity: alpha, beta and gamma diversity as described by R. H. Whittaker (Whittaker, 1960, 1972). Alpha diversity is defined as the local diversity or the diversity of a specific site and can be measured by examining species richness. Gamma diversity is the regional diversity over a larger area, which can be more difficult to determine. Beta diversity is the different of species compositions across different sites or a proportional relationship between alpha and gamma diversity (Whittaker, 1960, 1972). This study focuses on the alpha and beta diversity of avian communities within Acadia National Park. Although the simplest method of calculating beta diversity is the one described by Whittaker (1960, 1972), there are several other metrics that can be used and little consensus on which one is the best to use (Koleff *et al*, 2003; Tuomisto, 2010). Beta diversity can be calculated both using presence-absence values or abundance values (Barwell *et al*, 2015; Koleff *et al,* 2003). For this study, beta diversity was measured using presence-absence values. This was done to increase the comparability between two different data sets, as well as to account for variations in data collection in the Christmas Bird Count data. For presence-absence values, there are 24 different similarity and dissimilarity metrics for calculating beta diversity (Koleff *et al*, 2003). Each metric varies in how it accounts for different components such as nestedness, turnover, gradient usage and additive values (Koleff *et al,* 2003).

The three primary indices used in this study to measure both temporal and spatial beta diversity are Sørensen (sor), Jaccard (j), and Whittaker (w). I choose these three indices because they use the same scale for measurement, and account for continuity, symmetry and homogeneity (Koleff *et al,* 2003). These measures account for continuity by focusing on the species that are shared between two study quadrants. It is important to use symmetric measures to calculate beta diversity because the results should remain unchanged if neighboring and focal quadrant are switched (Koleff *et al,* 2003). Homogeneity allows that if the matching and nonmatching components are multiplied by a constant, the beta diversity values are the same (Koleff *et al,* 2003). All three of these indices use quadrants of time or space to measure beta diversity. Two of the indices, Sørensen and Jaccard, are similarity measures, meaning that the closer the value is to one, the more similar the two quadrants are. The opposite is true for Whittaker in that the closer the value is to one, the greater diversity between the two quadrants. Whittaker index is the most commonly used index for calculating beta diversity and represents the 'true' beta diversity as defined by Whittaker in 1960 (Koleff *et al,* 2003).

Statistical Analyses

Temporal changes in biodiversity

I conducted the statistical analysis for this research using Microsoft Excel and Program R (R Core Team, 2017). Originally collected as abundance data, the Christmas Bird Count Data was converted to presence-absence data where one denoted presence and zero denoted an absence. This was done to account for discrepancies in the abundance counts that could have been caused by different methods of counting as well as different-sized count groups within the Christmas bird count data set as well as in between the Christmas Bird Count dataset and the Schoodic Ecosystem Services Project dataset.

I used a linear regression (lm command, R base package) to test whether alpha diversity in the avian communities, as measured by species richness, of Mount Desert Island and Schoodic Peninsula have changed over time. I modeled species richness as a function of two predictive covariates, year and location (Mount Desert Island vs. Schoodic Peninsula), to understand both the temporal and spatial affects. The models included measuring the impact of year and location, in addition to a separate additive combination of year and location, and an interaction between year and location.

Because Christmas Bird Count methods have changed throughout time, notably, in 1950 and because surveys on Mount Desert Island and Schoodic Peninsula didn't start in the same year, I ran all liner regression tests on three different time series with varying start dates. In addition to calculating the liner regression trends from the start of both sets of data (1933 for Mount Desert Island and 1956 for Schoodic Peninsula), I conducted separate tests on the data from 1956 to 2016 and 1960 to 2016. I predicted that the change in methodology might have influenced the species richness over time. I selected the best model using Akaike's Information Criterion (AIC), *P*-value and adjusted R-squared (R^2) values. Akaike's Information Criterion examines the quality of statistical tests using the same data set. The lower AIC value represents the better model, and models that differ by more than two in AIC values are considered significantly different (Burnham and Anderson, 2002). It only gives the relative quality of a model, but does not address the absolute value. If all of the models are poor representations of the data AIC will only be able to say which one is better, but it still might not be an effective representation (Burnham and Anderson, 2002). Therefore, I used both *P*-values and adjusted R^2 values to assess fit of top models.

I calculated beta diversity for the temporal dataset between Schoodic Peninsula and Mount Desert Island using the vegan package (Oksanen *et al,* 2017). I calculated the beta diversity using the betadiver function which produced a distance matrix with all the calculated beta diversities. I calculated the beta diversity observed in each year compared to the species pool observed across all years using all 24 presence-absence indices for the MEMD and MESP data sets using the betadiver function. The average was taken of the matrix and represented the average beta diversity. This can be used as a measure for beta diversity when there isn't an exact distance or time scale to use. However, in this experiment, a time continuum was known, so the measured beta diversity was analyzed against the number of years since 1960 and a linear regression was completed. All of the initial start years were tested using alpha diversity and demonstrated similar patterns. The year 1960 was used as a representation of all years, and was ten years after the start of Schoodic Peninsula Christmas Bird Count Data.

Statistical analyses were completed directly comparing Schoodic Peninsula and Mount Desert Island each year to calculate beta diversity. Every five years starting in 1960, I calculated the average beta diversity values using Sørensen, Jaccard and Whittaker metrics between MEMD and MESP (betadiver command, vegan package). I then completed a linear regression analyzing the result of the beta diversity index to the number of years since 1960 for Sørensen, Jaccard and Whittaker indices and justified using slope, *P* values and adjusted R^2 values.

Spatial changes in biodiversity

Similar statistical analyses were used to measure the spatial changes in biodiversity. The SchESP abundance data was converted to presence-absence data where one represented the presences of a species and zero represented the absence of a species. I used a linear regression (lm command, R base package) to test whether alpha diversity in the avian community, as measured by species richness, has changed based on distance from an edge of human development. The models included measuring the impact of distance from closest edge, edge type, an additive combination of distance and edge type, and an interaction between distance and edge type. To compare whether the interactive or additive model was a better representation, AIC values were calculated in addition to *P* values, adjusted R^2 values and F statistics to assess model fit.

Once it was determined that edge type and distance from edge followed an interactive relationship to predict species richness, the individual edge types were analyzed in a linear fashion to determine how the distance from each specific edge type affected the species richness of the area. This was done by using the output of the linear regression of the interactive model to calculate the trend lines of each edge type.

In addition to calculating alpha diversity, I calculated beta diversity using the R Program vegan package (Oksanen *et al,* 2017). I calculated the average beta diversity observed in each point count location compared to the species pool across all locations. I performed a linear regression relating beta diversity indices and the total distance (m) from the edge. I measured beta diversity values for the SchESP data set only using Sørensen, Jaccard, and Whittaker indices.

RESULTS

Temporal Changes in Biodiversity

Alpha diversity, or local species richness, increased over time for both Mount Desert Island and Schoodic Peninsula using Christmas Bird Count data. For Mount Desert Island, using linear analysis, there was an increase in species richness by 0.24 species per year ($F_{1, 73}$=18.18, P= < 0.001, adjusted R^2=0.19). For Schoodic Peninsula, using linear analysis, there was an increase in species abundance of 0.25 species per year ($F_{1, 56}$=12.58, P= < 0.001, adjusted R^2=0.17). The greatest increase of species richness on Mount Desert Island was observed between 1951 and 1955, when there was an increase from 36 species to 58 species, consistent with records of changes in methodology. The increase followed a slight decline in species richness around the same time that Acadia National Park would have been affected by the 1947 fire. After the significant jump in species richness, it appears that the overall species richness slightly declines or levels out. I was unable to find any specific events that might pertain to the changes during this time period. Overall, Mount Desert Island has a higher number of observed species throughout time, but the species increased at relatively equal rates at Mount Desert Island and Schoodic Peninsula. After an initial increase, the species abundance on Schoodic Peninsula appears to level out and remain relatively the same.

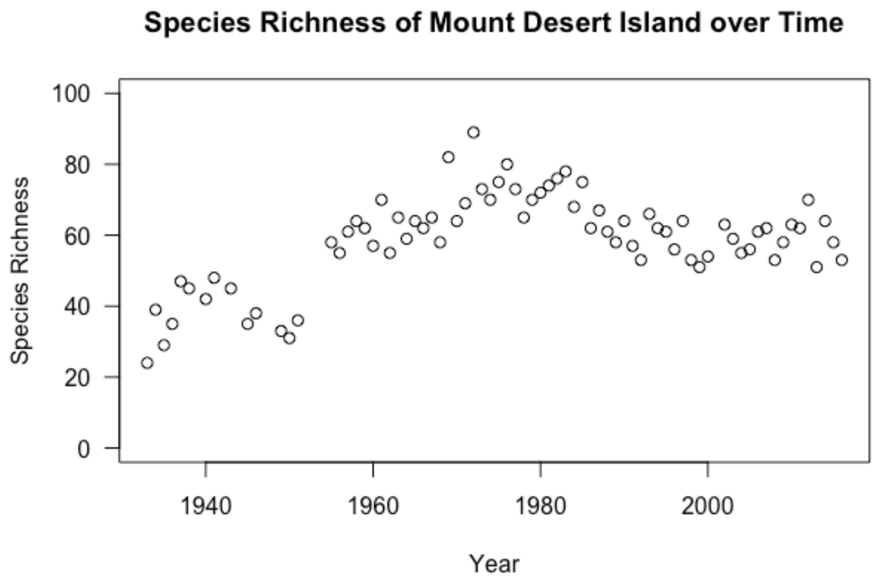

Figure 8 Change in avian species richness on Mount Desert Island from 1933 as represented in Christmas Bird Count data

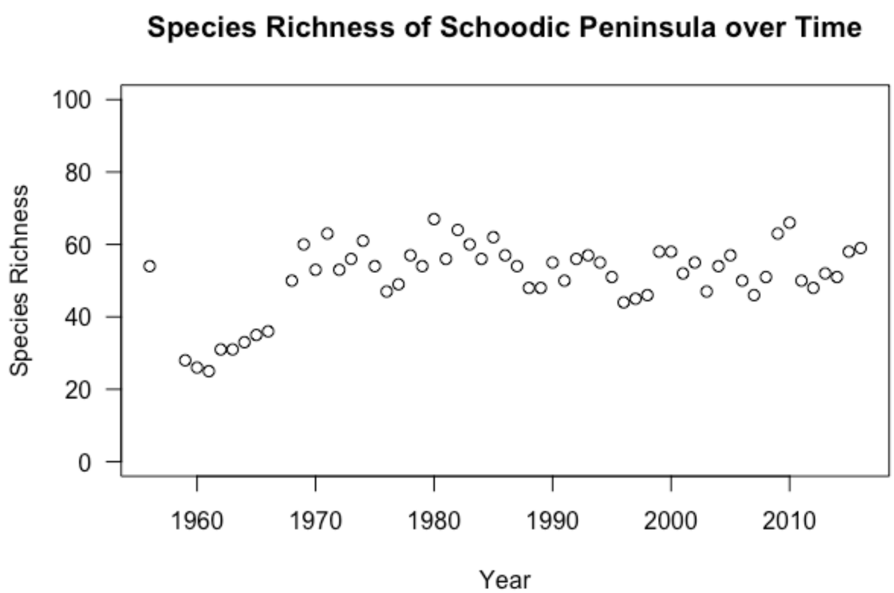

Figure 9 Change in avian species richness of Schoodic Peninsula from 1956 as represented in Christmas Bird Count

Of the candidate model set, an additive combination of year and location significantly predicted species richness ($F_{10.53,\ 130}$=24.57, P= < 0.001, adjusted R^2=0.26). Across both sites, species richness has increased at a rate of 0.243 per year, and approximately 10 fewer species per year were observed on Schoodic Peninsula relative to Mount Desert Island from 1933 to 2016.

Predictors of Species Richness (1933 to 2016)	AIC	Adjusted R^2	*P*-value	F-Statistic	Negative Log Likelihood	K
Year + location	1008.66	0.26	< 0.001	24.57	500.33	4
Year * Location	110.66	0.26	< 0.001	16.25	500.33	5
Year	1033.68	0.10	< 0.001	16.32	513.84	3
Location	1035.42	0.09	< 0.001	14.41	514.71	3

Table 2 Comparison of temporal species richness models relating year, 1933 to 2016, to location, Mount Desert Island or Schoodic Peninsula for the model selection process

Species richness predictors were also measured for 1956, and 1960 in an additive and multiplicative interaction between Mount Desert Island and Schoodic Peninsula. All trends showed increasing species richness to different values of significance and trend patterns. An additive combination of year and location significantly predicted species richness for the dataset starting in 1933, while a multiplicative combination of year and location significantly predicted species richness for the datasets starting in 1956 and 1960. The results from the 1933 dataset are included above to represent the start of the CBC data. However, the results from the 1956 and 1960 data sets are located in appendix C. All subsequent analyses for beta diversity were calculated starting in the year 1960.

The table below lists the corresponding average temporal beta diversity values for each of the metrics for calculating beta diversity. However, the focus was Sørensen (sor), Jaccard (j) and Whittaker (w). The mathematical calculations used for each metric are listed in appendix A. There is a wide variation in the averages, due to the differences in calculations and some metrics measure similarity and some measure dissimilarity. Overall, most indices indicated that the species pool in any given year was moderately similar to the species pool across all years. There was slightly more similarity at Mount Desert Island than Schoodic Peninsula for the species pool in any given year compared to the overall year. This could represent higher overall species richness levels or less change. The Sørensen similarity index, as well as the Jaccard and Whittaker indices were three that demonstrated moderate similarity through time for both Schoodic Peninsula (0.717, 0.563, 0.283) and Mount Desert Island (0.739, 0.589, 0.261). The table below serves as a resource to understanding the variety of beta diversity indices.

Index	MESP	MEMD
Whittaker (β_w)	0.283	0.261
Harrison (β_{-1})	0.282	0.261
Cody (β_c)	14.768	17.090
Weiher and Boylen (β_{wb})	29.537	34.180
Routledge's R (β_r)	0.085	0.085
Routledge's I (β_I)	0.186	0.177
Routledge's E (β_e)	0.206	0.195
Wilson and Schmida (β_t)	0.283	0.261
Mourelle and Ezcurra (β_{me})	0.283	0.261
Jaccard (β_j)	0.563	0.589
Sørensen (β_{sor})	0.717	0.739
Magurran (β_m)	45.771	53.854
Harrison (β_{-2})	0.169	0.181
Cody (β_{co})	0.270	0.256
Colwell and Coddington(β_{cc})	0.437	0.411
Gaston (β_g)	0.437	0.411
Williams (β_{-3})	0.142	0.151
Lande (β_l)	14.768	17.090
Williams (β_{19})	0.079	0.079
Harte and Kinzig (β_{hk})	0.283	0.261
Ruggiero (β_{rlb})	0.763	0.723
Simpson (β_{sim})	0.198	0.204
Lennon (β_{gl})	0.203	0.139
Lennon (β_z)	0.358	0.333

Table 3 Average temporal beta diversity of Mount Desert Island (MEMD) and Schoodic Peninsula (MESP) from 1960 to 2016 using all presence absence metrics with Christmas Bird Count Data

Temporal beta diversity was calculated using Sørensen and Jaccard similarity indices and Whittaker dissimilarly index for Schoodic Peninsula over time since 1960. All three indices for Schoodic Peninsula showed similar patterns. The dissimilarity of the avian community increased annually compared to the avian community observed in the first year, 1960. For Schoodic Peninsula there was a 0.001 decrease in beta diversity per year using Sørensen similarity index, ($F_{1, 52}$=6.387, P= 0.015, adjusted R^2=0.09), a 0.001 decrease in beta diversity per year using Jaccard similarity index ($F_{1, 52}$=7.175, P= 0.010, adjusted R^2=0.10), and a 0.001 increase in beta diversity per year using Whitaker dissimilarity index ($F_{1, 52}$=6.387, P= 0.015, adjusted R^2=0.09).

Index	Annual trend (beta diversity/year)	*P*-value	Adjusted R^2	Correlation
Sørensen (similarity)	-0.001	0.015	0.09	-0.33
Jaccard (similarity)	-0.001	0.010	0.10	-0.35
Whittaker (dissimilarity)	0.001	0.015	0.09	0.33

Table 4 Sorensen, Jaccard and Whittaker beta diversity trends compared to years since 1960 with correlation factors for Schoodic Peninsula

Temporal beta diversity was calculated using Sørensen and Jaccard similarity indices and Whittaker dissimilarly index for Mount Desert Island over time since 1960. All three indices for Mount Desert Island showed very statistically significant similar patterns. The dissimilarity of the avian community increased annually compared to the avian community observed in the first year, 1960. For Mount Desert Island there was a 0.003 decrease in beta diversity per year using Sørensen similarity index, ($F_{1, 52}$=161.3, P= < 0.001, adjusted R^2=0.75), a 0.004 decrease in beta diversity per year using Jaccard similarity index ($F_{1, 52}$=167.1, P= < 0.001, adjusted R^2=0.76), and a 0.003 increase in beta diversity per year using Whitaker dissimilarity index (($F_{1, 52}$=161.3, P= < 0.001, adjusted R^2=0.75).

Index	Annual trend (beta diversity/year)	*P*-value	Adjusted R^2	Correlation
Sørensen (similarity)	-0.003	< 0.001	0.75	-0.87
Jaccard (similarity)	-0.004	< 0.001	0.76	-0.87
Whittaker (dissimilarity)	0.003	< 0.001	0.09	0.87

Table 5 Sorensen, Jaccard and Whittaker beta diversity trends compared to years since 1960 with correlation factors for Mount Desert Island

The results from analyzing beta diversity over time for both Schoodic Peninsula and Mount Desert Island both present increasing dissimilarity of the avian community compared to the avian community observed in the first year, 1960 (Figures 10 – 15).

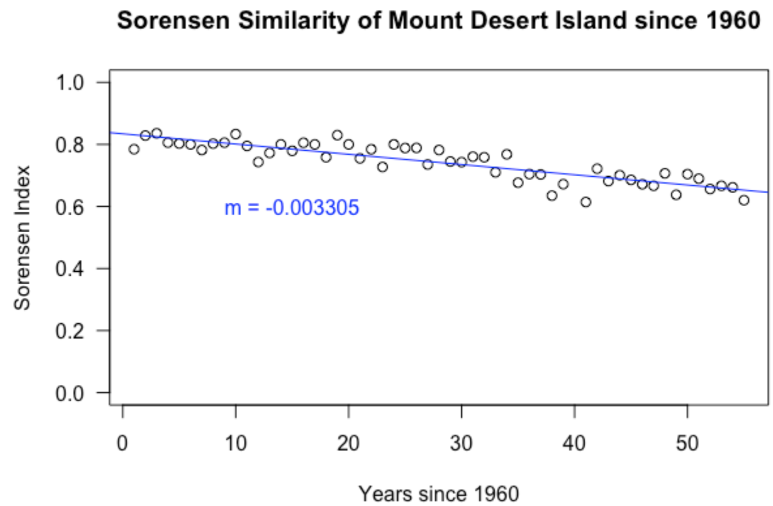

Figure 10 Sørensen similarity index of Mount Desert Island (1960 to 2016) vs number of years since 1960

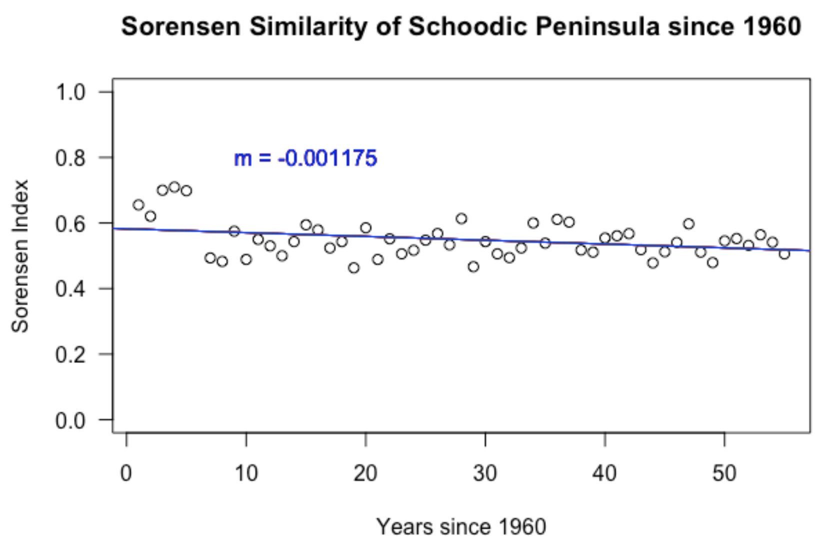

Figure 11 Sørensen similarity index of Schoodic Peninsula (1960 to 2016) vs number of years since 1960

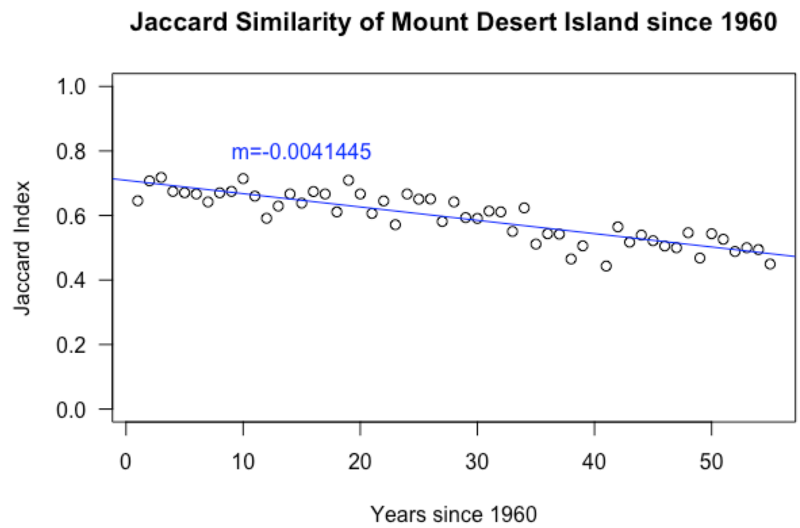

Figure 12 Jaccard similarity index of Mount Desert Island (1960 to 2016) vs number of years since 1960

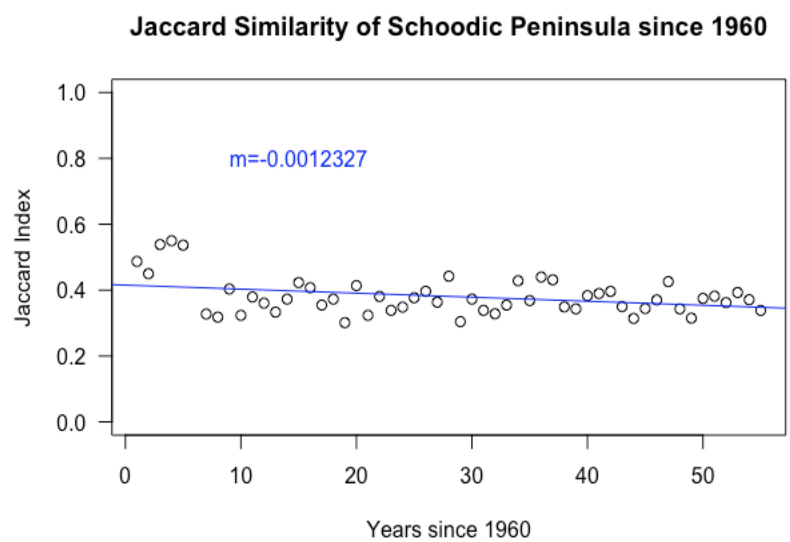

Figure 13 Jaccard similarity index of Schoodic Peninsula (1960 to 2016) vs number of years since 1960

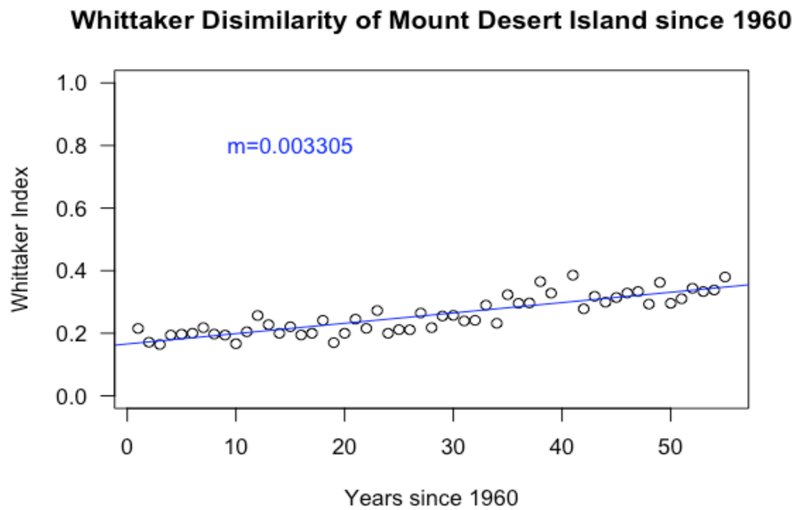

Figure 14 Whittaker dissimilarity index of Mount Desert Island (1960 – 2016) vs number of years since 1960

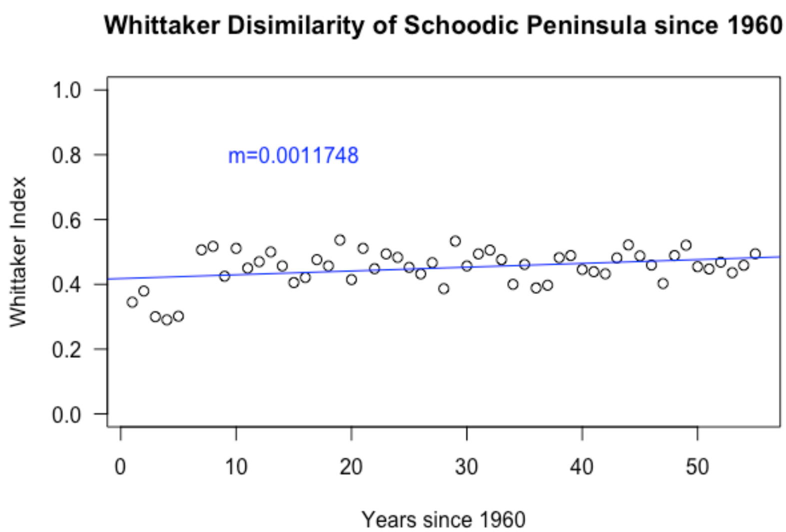

Figure 15 Whittaker dissimilarity of Schoodic Peninsula (1960 – 2016) vs number of years since 1960

The average value of beta diversity using Sørensen similarity, Jaccard similarity, and Whittaker dissimilarity indices were calculated comparing Mount Desert Island and Schoodic Peninsula every five years starting in 1960. Patterns were consistent across the Sørensen, Jaccard and Whittaker indices in that they all demonstrated that Mount Desert

Island and Schoodic Peninsula are becoming more similar over time. The average beta diversity for each year is included in the table below. The greatest value for Sørensen index was 0.819 in 2005 and the smallest was 0.602 in 1960. Beta diversity increased from 1960 to roughly 1980 where it dipped slightly, but returned to the same level around 2005. Figures 16 through 18 demonstrate this trend. Using the Sørensen index, the similarity between Schoodic Peninsula and Mount Desert Island increased linearly by 0.002 species per year ($F_{1, 10}$=7.652, P= 0.020, adjusted R^2=0.38). Using the Jaccard index the similarity between Schoodic Peninsula and Mount Desert Island increased linearly by 0.003 species per year ($F_{1, 10}$=7.464, P= 0.021, adjusted R^2=0.37). Using the Whittaker index, the dissimilarity between Schoodic Peninsula and Mount Desert Island decreased linearly by 0.002 species per year ($F_{1, 10}$=7.652, P= 0.020, adjusted R^2=0.38).

	Sørensen	Jaccard	Whittaker
1960	0.602	0.431	0.398
1965	0.667	0.500	0.333
1970	0.752	0.603	0.248
1975	0.760	0.613	0.240
1980	0.814	0.687	0.186
1985	0.759	0.612	0.241
1990	0.752	0.602	0.248
1995	0.741	0.589	0.259
2000	0.759	0.611	0.241
2005	0.819	0. 693	0.181
2010	0.803	0.671	0.197
2015	0.759	0.611	0.2414

Table 6 Sørensen, Jaccard and Whittaker indices of Mount Desert Island compared to Schoodic Peninsula every five years since 1960

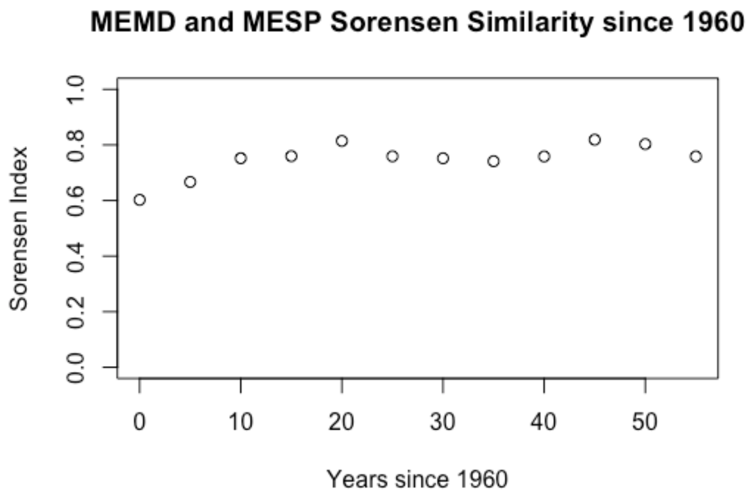

Figure 16 Sørensen similarity index compared across Mount Desert Island and Schoodic Peninsula (1960 – 2016) vs number of years since 1960

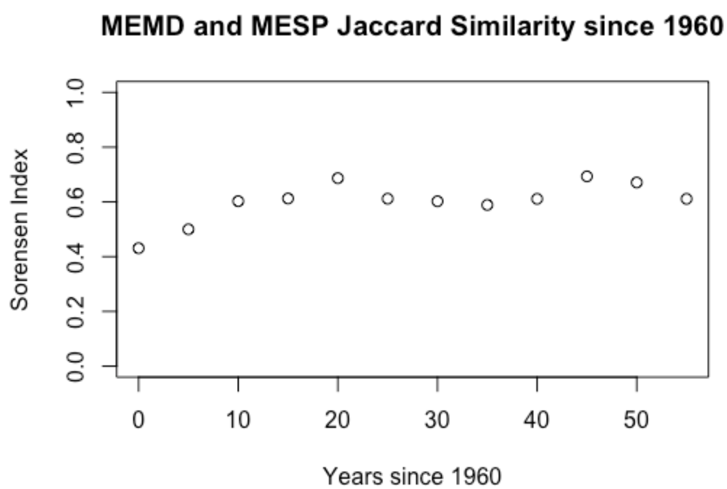

Figure 17 Jaccard similarity index compared across Mount Desert Island and Schoodic Peninsula (1960 – 2016) vs number of years since 1960

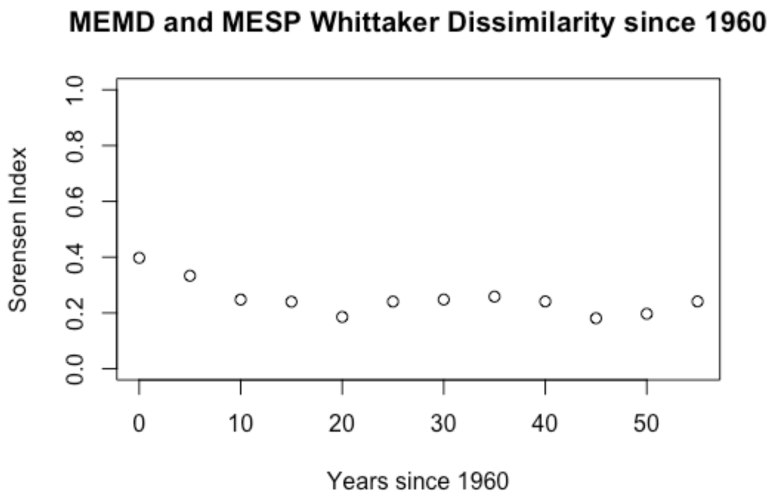

Figure 18 Whittaker similarity index compared across Mount Desert Island and Schoodic Peninsula (1960 – 2016) vs number of years since 1960

Spatial Changes in Biodiversity

Of the candidate model set, an interactive combination of distance from closest edge and edge type best predicted species richness ($F_{2.187,\,38}$=8.54, *P*= <0.001, adjusted R^2=0.51). Edge type was a categorical variable and distance from closest edge was a numerical variable measured in meters.

Predictors of Species Richness	AIC	Adjusted R^2	P – value	F statistic	Negative log Likelihood	K
Distance from closest edge + edge type	214.10	0.39	<0.001	8.15	101.05	6
Distance from closest edge * edge type	206.51	0.51	<0.001	8.54	95.25	8
Distance from closest edge	231.85	0.04	0.090	3.03	112.92	3

Table 7 Comparison of spatial species richness models relating distance from closest edge type in the area surrounding Schoodic Woods to the edge type (i.e. campground or bike path) for model selection process

After determining that the effect of distance from the closest edge and edge type are dependent on each other in determining species richness, the individual trends relating distance and edge were measured. Overall, species richness decreased as the distance increased for the edge types bike path, and campground. Species richness increased as the distance increased for the edge type road. There was only one point count with ocean as the closest edge so it was not included in the analysis. The bike path demonstrated a 0.011 decrease ($F_{1, 12}$=9.40, P= 0.010, adjusted R^2= 0.39) in the number of species per meter from the edge. The campground edge demonstrated a 0.010 decrease ($F_{1, 11}$=3.15, P= 0.103, adjusted R^2=0.15) in the number of species per meter from the edge. The road edge demonstrated a 0.002 increase ($F_{1, 15}$=1.19, P=0.292, adjusted R^2=0.01) in the number of species per meter form the edge. Besides the road, there was greater species diversity near the edges as can be observed in figure 16 below. The campground had the greater initial number of species followed by the bike path, and finally the road.

Figure 19 Species richness of the area surrounding the Schoodic Woods Campground compared to the distance from closest edge and edge type

The graph below depicts the relationship between beta diversity (Sørensen similarity index) and the distance from the closest edge in the area surrounding the Schoodic Woods Campground. There was no significant linear relationship (*P*=0.727, adjusted R^2=0.02) between these two variables. It is predicted that similar tests using Jaccard similarity index and Whittaker dissimilarity index would demonstrate similar trends.

Figure 20 Spatial Sørensen similarity index of Schoodic Peninsula vs distance from the closest edge with edge type noted with color differentiation

DISCUSSION

Temporal changes in biodiversity

Biodiversity is an important measure of ecosystem integrity. Studies have shown that high levels of diversity allow for multifunctional, resilient and healthy ecosystems (Gamfeldt *et al,* 2008; Perkins *et al,* 2014). Loss of biodiversity can also lead to decreases in habitat integrity.

Both of the alpha diversity and beta diversity demonstrated patterns consistent with increases in species diversity. The alpha diversity, or species richness, for both Mount Desert Island and Schoodic Peninsula increased over time across all time periods examined (1933, 1956 or 1960 to 2016). The time series beginning in 1933 was best described by an additive combination of location and year, while the 1956 and 1960 datasets exhibited a multiplicative trend between the two variables. In general, a higher level of alpha diversity would indicate a more resilient ecosystem (Gamfeldt *et al,* 2008; Perkins *et al,* 2014). However, additional studies have noted that an increase in alpha diversity can occur as generalists become more prevent in the ecosystem compared to specialists (Barnagaud *et al,* 2011; Davey *et al,* 2012). Depending on the situation, an increase in generalist species, and a corresponding increase in alpha diversity, is not always preferred and doesn't necessarily mean a more resilient ecosystem. This particular study didn't test the effects of specific drivers or examine the specific changes in species abundance, so while there might be correlation, the specific cause and differentiation of the changes isn't known.

Examining temporal beta diversity for both Schoodic Peninsula and Acadia National Park demonstrated that within each area there is more dissimilarity in populations as time increases compared to the original examined population in 1960. The year 1960 was used as a refence point because it was after the change in methodology in Christmas Bird Count data and demonstrated significant results. This supports the hypothesis that the diversity in each location is changing and correlates with the increase in alpha diversity. Various studies have demonstrated that this pattern could correspond to increases in anthropogenic impact and climate change (Matthews, 2014; Olden, 2006; Savage and Vellend, 2014; Wilson *et al,* 2015;). Although the increase in dissimilarity throughout time demonstrates that the avian communities are changing, it isn't possible to draw conclusions regarding stressors or the size of the change.

The beta diversity values calculated between Schoodic Peninsula and Mount Desert Island over time demonstrated increased similarity. This shows that Schoodic Peninsula is becoming more like Mount Desert Island. This corresponds to the predicted patterns. Schoodic Peninsula has recently become more developed and less forested. This could be a contributing factor as to why the two locations have increased in similarity. Climate change operates on a global scale and it is possible that the combination of close proximity between the two locations and global warming could lead to increased similarity.

<u>Spatial changes in biodiversity</u>

The spatial alpha diversity trends exhibited a multiplicative relationship between distance from the closest edge and edge type with the greatest number of species closer to the edges. In addition, all of the edge surfaces demonstrated increases in biodiversity closer to the edge except for the road. The ocean edge type was not used in analysis due to only having one point. This would suggest that anthropogenic impact, in the form of increased edges and habitat fragmentation, impacts avian richness. Specifically, this could be due to edge effect, the change in community structures where two different habitats meet, and the rise of generalist species in disturbed areas (Barnagaud *et al,* 2011; Davey *et al,* 2012). One study done by Ortega and Capen found that unpaved roads through forest acted as an edge for avian species and abundance was impacted by the road (Ortega and Capen, 2002). The decrease in species richness in correlation to the decrease in distance from the road edge could be due to the wider, more traffic heavy surface.

Comparing beta diversity to the distance from the closest edge in the area surrounding Schoodic Woods campground did not return a significant trend. Although, studies examining spatial beta diversity are less common than studies examining temporal beta diversity, this does not align with the recorded trends. One study noted that beta diversity could increase due to habitat fragmentation or decrease due to human disturbance, but there was an observed trend (Socolar *et al,* 2016). The variance could be due to using a metric of beta diversity that didn't fit the data source or that there were multiple edges affecting the beta diversity compositions. Further studies are needed to understand and expand upon these results.

Conservation Implications

Understanding biodiversity can aid scientists in understanding ecosystem functioning and lead to improved policies and conservation efforts. For example, beta diversity can be used to predict gamma diversity patterns, quantify biodiversity loss, inform placement of protected areas, help manage biological invasions, and protect rare species (Socolar *et al,* 2016). However, there is a lot of confusion and varied opinions on the proper indices of beta diversity, how to analyze the data and the impacts of various stressors on beta diversity (Anderson *et al,* 2010; Koeff *et al,* 2003; Socolar *et al,* 2016). Maximum beta diversity is not necessarily desirable for maintain or increasing gamma diversity, the overall species diversity (Socolar *et al,* 2016). One study done on agriculture development, found that maintaining the levels of beta diversity was correlated with a positive outcome on overall diversity and ecosystem integrity (Joana *et al,* 2017). Beta diversity is a complicated subject and that affect the decisions that can be made based on it. Although the number of studies focusing on beta diversity is rapidly increasing, beta diversity is rarely used as the primary biodiversity factor in making management and protection decisions due to its complicated nature (Joana *et al, 2017*). Preliminary studies have linked decreased beta diversity to biological invasions, urbanization, and potentially climate change and anthropogenic impact (Socolar *et al,* 2016). However, these patterns have been variable and little research has been done linking climate change and anthropogenic impact to trends in beta diversity.

Measuring biodiversity using beta diversity has the potential to aid researchers and policy makers with future conservation decisions, especially with the various methods of calculating beta diversity and measuring it. However, additional studies are needed to set baseline patterns for beta diversity effects in response to stressors such as climate change, anthropogenic impact, habitat fragmentation, agriculture and urbanization. Additional research is also needed to understand how each index can be applied and statistically analyzed. The variance in indices may not be a problem as some researchers have believed, but it may provide the opportunity for more specialized analysis (Tuomisto, 2010). However, using beta diversity measures in addition to other models can aid in overall understanding (Gering *et al,* 2003). Beta diversity is a useful measure of biodiversity, but we need to have a better understanding of it in order to use it to its full potential.

Although this study provides limited conclusions regarding the overall understanding of beta diversity, it has demonstrated change in the avian communities within Acadia National Park. The alpha diversity of both Mount Desert Island and Schoodic Peninsula increased over time and the beta diversity increased as well. This preliminary study of beta diversity between Mount Desert Island and Schoodic Peninsula demonstrated that there were becoming slightly more similar. The case study examining spatial beta diversity surrounding Schoodic Woods was inconclusive, but there was a relationship for alpha diversity between distance from the edge and edge type. Additional studies are needed to further understand the beta and alpha diversity trends of the avian communities of Acadia National Park, but this study indicates there is a potential for using beta diversity in conservation decisions.

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APPENDICES

APPENDIX A: BETA DIVERSITY PRESENCE-ABSENCE INDICES EQUATIONS

```
> betadiver(help=TRUE)
1 "w" = (b+c)/(2*a+b+c)
2 "-1" = (b+c)/(2*a+b+c)
3 "c" = (b+c)/2
4 "wb" = b+c
5 "r" = 2*b*c/((a+b+c)^2-2*b*c)
6 "I" = log(2*a+b+c) - 2*a*log(2)/(2*a+b+c) - ((a+b)*log(a+b) +
(a+c)*log(a+c)) / (2*a+b+c)
7 "e" = exp(log(2*a+b+c) - 2*a*log(2)/(2*a+b+c) - ((a+b)*log(a+b) +
(a+c)*log(a+c)) / (2*a+b+c))-1
8 "t" = (b+c)/(2*a+b+c)
9 "me" = (b+c)/(2*a+b+c)
10 "j" = a/(a+b+c)
11 "sor" = 2*a/(2*a+b+c)
12 "m" = (2*a+b+c)*(b+c)/(a+b+c)
13 "-2" = pmin(b,c)/(pmax(b,c)+a)
14 "co" = (a*c+a*b+2*b*c)/(2*(a+b)*(a+c))
15 "cc" = (b+c)/(a+b+c)
16 "g" = (b+c)/(a+b+c)
17 "-3" = pmin(b,c)/(a+b+c)
18 "l" = (b+c)/2
19 "19" = 2*(b*c+1)/(a+b+c)/(a+b+c-1)
20 "hk" = (b+c)/(2*a+b+c)
21 "rlb" = a/(a+c)
22 "sim" = pmin(b,c)/(pmin(b,c)+a)
23 "gl" = 2*abs(b-c)/(2*a+b+c)
24 "z" = (log(2)-log(2*a+b+c)+log(a+b+c))/log(2)
```

APPENDIX B: R SAMPLE COMMANDS

Shortened Code List

```
summary()
AIC()
read.csv(file.choose())
lm1=lm(Species.Richness ~ Year + Locations, data = CBC)
lm2=lm(Species.Richness ~ Year * Locations, data = CBC)
CBC.1956 = subset(CBC, Year > 1956)
lm5=lm(Species.Richness ~ Year + Locations, data = CBC.1956)
lm6=lm(Species.Richness ~ Year * Locations, data = CBC.1956)
CBC.1960 = subset(CBC, Year > 1960)
lm7=lm(Species.Richness ~ Year + Locations, data = CBC.1960)
lm8=lm(Species.Richness ~ Year * Locations, data = CBC.1960)
ln1=lm(Spp.Richness ~ DistFromClosestEdge + ClosestEdgeType, data = SCH)
ln2=lm(Spp.Richness ~ DistFromClosestEdge * ClosestEdgeType, data = SCH)
MESPBA=read.csv(file.choose())
MEMDBA=read.csv(file.choose())
mean(betadiver(MESPBA, "w”)) - done for all indices
mean(betadiver(MEMDBA, "w”)) - done for all indices
MESPS=(betadiver(MESPBA, "sor"))
MEMDS=(betadiver(MEMDBA, "sor"))
m=as.matrix(MESPS)
m1=m[,1]
m2=as.vector(m1)
m3=m2[-1]
m4=0:55
m5=m4[-c(1,7)]
m24=data.frame(m5,m3)
plot(m24)
M=plot(m24, main = "Sorensen Similarity of Schoodic Peninsula since 1960", xlab= "Years since 1960",
ylab="Sorensen Index", las=1, ylim=c(0,1))
abline(lm(m3~m5), col=4)
cor(m5,m3)
summary(lm(m3~m5))
AMD=read.csv(file.choose(), header = TRUE)
AMD1=plot(AMD, main = "Species Richness of Mount Desert Island over Time", xlab= "Year",
ylab="Species Richness", las=1, ylim=c(0,100))
ASP=read.csv(file.choose(), header = TRUE)
DIS=read.csv(file.choose())
DIS1=(betadiver(DIS, "sor"))
DI=as.matrix(DIS1)
D1=DI[,1]
D2=as.vector(D1)
ED=read.csv(file.choose(),header=FALSE)
ED1=as.vector(ED)
ED2=ED1[-c(1)]
EDIS=data.frame(ED1,D2)
mean(betadiver(y60, "j”)) - repeated for years 1960 - 2015
mean(betadiver(y60, "sor”)) - repeated for years 1960 - 2015
mean(betadiver(y60, "w”)) -repeated for years 1960 – 2015
```

APPENDIX C: SPECIES ABUNDANCE PREDICTORS FOR 1956

Of the 1956 candidate model set, a multiplicative combination of year and location significantly predicted species richness ($F_{8.269,\ 112}$=31.99, P=5.162e-15, adjusted r^2=0.447).

Predictors of Species Richness (1933 to 2016)	AIC	Adjusted R^2	*P*-value	F-Statistic
Year + location	846.5069	0.3302	5.433e-11	29.35
Year * Location	825.2357	0.4471	5.162e-15	31.99
Year	893.0052	-0.009	0.8674	0.028
Location	844.8351	0.3342	5.517e-12	58.73

APPENDIX D: SPECIES ABUNDANCE PREDICTORS FOR 1960

Of the 1960 candidate model set, a multiplicative combination of year and location significantly predicted species richness ($F_{7.93,\ 106}$ =29.64, P=5.383e-14, adjusted r^2=0.4408).

Predictors of Species Richness (1933 to 2016)	AIC	Adjusted R^2	*P*-value	F-Statistic
Year + location	793.117	0.3265	2.437e-10	27.42
Year * Location	773.6227	0.4562	5.383e-14	29.64
Year	836.3034	-0.00628	0.5732	0.3193
Location	791.4267	0.3308	2.99e-11	54.89

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Marie Ring was born in Farmington, Maine on May 13, 1996. She was raised in Topsham, Maine and graduated from Mt. Ararat High School in June 2014. She is a senior majoring in biology and minoring in ecology and environmental sciences. Upon graduation, Marie plans to take a gap year to work and travel, and then pursue environmental research or outreach.